Sharing is caring - the importance of capital goods when assessing environmental impacts from private and shared laundry systems in Sweden

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BUILDING COMPONENTS AND BUILDINGS

Sharing is caring - the importance of capital goods when assessing environmental impacts from private and shared laundry systems in Sweden

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Abstract

Purpose Previous studies on environmental impacts from domestic laundry have tended to focus solely on private washing machines and detergent. However, public procurement guidelines about the construction of laundry spaces may also be important. This article aims to expand the scope of previous work so that it also includes tumble drying and the building space. By doing this, we examine the potential for shared systems (which are common in Sweden) to reduce the environmental impacts of laundry activities, in comparison with consumer choices associated with machine operation (i.e., wash temperature and amount of detergent).

Methods An LCA model was created using product information data from the European Union. Emissions from building use were taken from Swedish cradle-to-grave reports on energy-efficient buildings. The resulting model was run with additional sensitivity analysis of the variables, and the associated emissions from each of the scenarios were calculated.

Results and discussion On average, greenhouse gas (GHG) emissions for private laundries in Sweden were estimated to be 190 g CO_2 eq./kg laundry (washed and dried). If a shared laundry was used instead, the resulting emissions decreased by approximately 26%. The greatest contribution to GHG emissions was the use of detergent (22–33% of total emissions), followed by capital goods (11–38% of total emissions).

Conclusion Deciding to construct shared laundries in newly built apartment buildings in Sweden, rather than in-unit machines, would reduce the emissions from domestic laundry for these tenants by approximately 26%. This is because materials used for manufacturing whitegoods, as well as the emissions associated with the building itself, play a much bigger role than previously thought. Additionally, since the cleaning efficiency of warm water and some of the components used in detergents rises with temperature, emissions from domestic laundering could for some consumers be reduced further by washing at *higher* temperature but with less detergent. This pattern could be seen in Sweden within regions with hard water, where the emissions from domestic laundry could be reduced by 6–12%.

Keywords Domestic laundry · Shared laundry · Life cycle assessment · Product service system · Community-based system · Household emissions · Sweden

1 Introduction

A recent study of 43 countries showed that household consumption causes more than 60% of global greenhouse gas (GHG) emissions and between 50% and 80% of total land, material, and water use (Ivanova et al. 2016). The most dominant categories were mobility, food, and shelter (including clothing consumption and household services). In Europe, domestic laundry on average consumes 4–9% of all the energy used in households, while at 17%, this value is notably higher in Turkey. Meanwhile, 8–12% of all the potable water

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consumed in households is used for washing clothes (Pakula and Stamminger 2010).

Traditional strategies for limiting environmental impacts from domestic laundry usually focus on technological advancements, for example automation of the laundering process (Pakula and Stamminger 2015). However, these types of improvements tend to coevolve with consumer practices (Shove 2003). A recent study of potential improvements for washing machines exemplifies this relationship. The authors concluded that even though several improvements are possible, the reduction of environmental impacts from laundry is hampered not by the possible technical improvements of the machines, but by the washing habits of the consumers (Laitala et al. 2011). Or in other terms, consumers of today buy more clothing (Peters et al. 2019) and wash it more frequently than during any other time in history (Klepp 2003). This means that even though today's washing machines can wash at lower temperatures, with less detergent, and with a larger amount of laundry than a decade ago (Graulich et al. 2011; Presutto et al. 2007a), the increased frequency of doing the laundry tends to counteract the technological improvements. In UK, for example, the overall energy associated with washing and drying clothes within households has grown by over 100% between 1970 and 2012 (Yates and Evans 2016). This is despite the fact that the average energy consumption for washing machines in Europe per cycle was cut by more than 50% over that time (Presutto et al. 2007a). This trend is not only limited to the Western world. For example, in China, the ownership of washing machines has grown from 5 to 97% in urban areas and from 0 to 62% in rural areas just during the years 1980–2012, making laundry easier (Wang et al. 2014).

One alterative way to reduce laundry impacts could be to move from private ownership of laundry products, to a community-based system (CBS) or a product service system (PSS) for laundry activities (Mont and Plepys 2007; Tukker 2015). Such alternative systems for laundry have been gaining popularity in circular economy thinking, with articles examining pay-per-use (Bocken et al. 2018) as well as customer preferences for owning, renting, or pay-per-use solutions for washing machines (Lieder et al. 2018). Since CBSs and PSSs (from here on called "shared systems") require fewer laundry machines, it is argued that these types of systems could limit environmental impacts, just as collaborative consumption of the clothes themselves can (Zamani et al. 2017). Previous studies have shown that such systems for domestic laundry could indeed reduce the environmental impacts by 30–50%, compared with private ownership (Garcilaso et al. 2007; Haapala et al. 2008). These findings are positive from an environmental point of view, but it should be noted the results are dependent on contextual factors. This means that resource consumption within both private and shared laundries varies considerably depending on culture, social factors, and availability (Retamal and Schandl 2018). In addition, previous published studies concerning shared systems for domestic laundry do not consider the changed need and material consumption associated with the building itself, or even the impacts associated with the supply of detergents and water (Borg and Hogberg 2014). However, a recent master's thesis from Linköping University indicated a possibility of lower environmental impacts associated with shared laundries in comparison with private ownership (Nilsson 2011). By designing buildings with a shared laundry room, less floorspace is needed per apartment, which in practice could mean that less material is used per building or that a higher number of apartments per building are feasible.

This article aims to extend previous studies by expanding the analytical system boundary to include the building space itself. We believe that by doing so, a more honest comparison can be made between private and shared laundries for domestic laundry. The first research question for the study is therefore: to what extent do capital goods (i.e., whitegoods and the building itself) influence the emissions associated with domestic laundry, and how does it vary between private and shared laundry systems?

It is worth noting that shared laundries are common in many countries, especially in more densely populated areas. For example, shared laundries (e.g., communal laundry rooms within a multi-family building, or a commercial coin-operated facility) are often used in the Philippines (Retamal and Schandl 2018), Japan and Thailand (Moon 2020) and Finland (Miilunpalo and Raisanen 2019). A recent study by Laitala et al. (2020) also found that the use of shared washing machines varied between 6 and 13% among consumers in China, Germany, Japan, UK, and the USA. For Sweden, shared laundries were introduced during the 1920s and have been the norm throughout the nation since the 1950s (Lund 2009). A major shift began in the early 1990s, and since then, most new multi-family buildings are equipped with private, in-unit washing machines, replacing the shared laundry rooms. This trend is also true for public housing in Sweden, which represent 46% of all Swedish rental apartments. However, since companies operating public housing have a stated mission to run the rental business in an environmental, economic and social sustainable way, this shift can be reversed by the public procurement process (Hall et al. 2016; Public Housing Sweden 2020). This paper is therefore intended as a contribution to a discussion about reversing laundry design norms in public housing in Sweden, in order to reduce emissions caused by domestic laundry.

In addition to investigating the influence of capital goods, this article aims to quantify and illustrate the potential trade-off between wash temperature and detergent dose. This dynamic is important to understand because of the ongoing initiatives in Europe that make consumers wash at lower temperatures (Laitala et al. 2012). One such example is

Procter & Gamble's "*low-temperature laundry initiative*" that recently developed into the ongoing international campaign "*I prefer 30*" (Mylan 2017). Unfortunately, the effectiveness of the components in many of today's detergents varies with temperature and lower wash temperatures can result in worse cleaning results. Consumers might thus be inclined to compensate the lower wash temperatures with higher amount of detergent, although it is unclear how this affects the overall emissions of a load of laundry. Therefore, the second research question for this paper is: how could net emissions potentially change in other EU countries on account of the tradeoff between temperature and detergent dose? Our main focus is on environmental impacts from Swedish domestic laundry. However, by also using data sources that are representative for European consumers, the results will indicate the possible magnitude of regional variations.

2 Methodology

2.1 Goal

The overall goal of this LCA is to examine the potential for shared systems to reduce the environmental impacts of laundry activities, in comparison with consumer choices associated with machine operation (i.e., wash temperature and amount of detergent). The functional unit for the study was the washing and drying of 1 kg of clothing. The choice of focusing on the practice rather than the different system components was made so that the results would be compatible with other life cycle studies, for example with focus on textile and clothing consumption (Sandin et al. 2019). This analysis is driven by curiosity rather than the needs of a particular decision-maker. However, the intended audience includes urban planners, architects, appliance manufacturers, and other actors interested in the sustainability of laundry systems.

2.2 Scope

The scope of the study is based on a cradle-to-grave system for the machines, the cleaning products (detergent, softener and bleach), and the part of the building used for the machines within the system. The system boundary for the LCA model is depicted in Fig. 1.

2.2.1 The role of capital goods in shared versus private systems

To fulfill the goal of the study, three attributional LCA models containing the three different systems (private laundry, shared laundry 1, and shared laundry 2) were created using the GaBi-software with datasets found in Ecoinvent 3.5 and GaBi Professional 2021. Data used in the models for the machines (e.g., bill of materials) was based on the European Commission reports on preparatory studies for Eco-design Requirements (EuP). The data was deemed appropriated since the reports give a thorough review of the European market for whitegoods, while at the same time distinguishing between private washing machines and dryers (Presutto et al. 2007a; PricewaterhouseCoopers 2009) and professional ones

Fig. 1 The system boundary of the LCA model

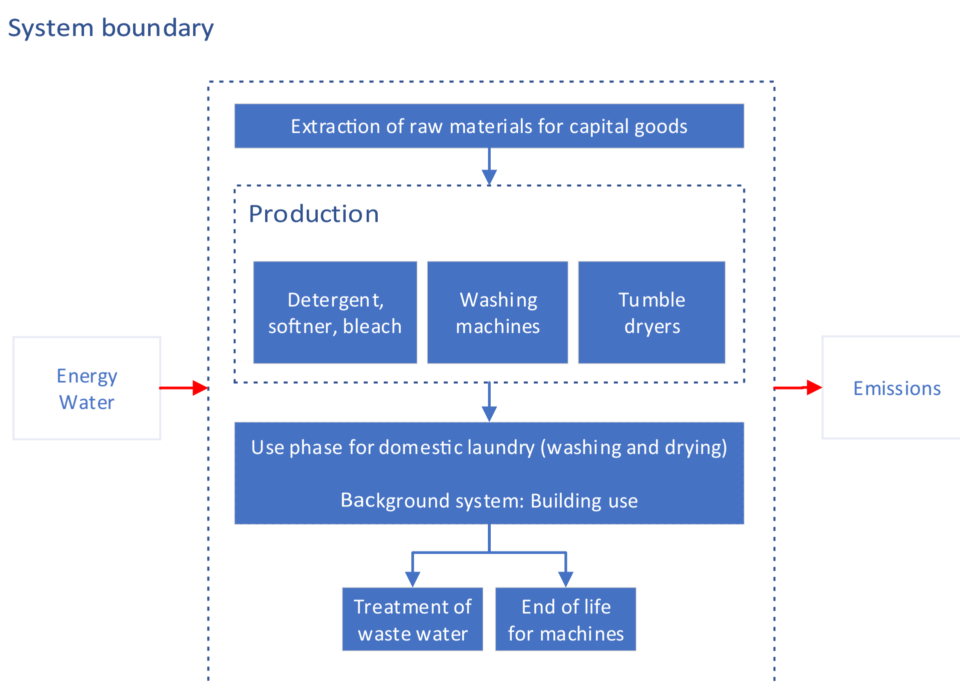

Table 1 Summary of the characteristics of each laundry system

Specifications	Private laundry	Shared laundry 1	Shared laundry 2
Washing machine load capacity	5 kg	6 kg	10 kg
Loading rate (amount of laundry)	60% (3 kg)	60% (3.6 kg)	60% (6 kg)
Type of washing machine	Private	Semi-professional	Professional
Average no. cycles during lifetime for washing machine	1 100	11 000	24 000
Tumble dryer size	6 kg	6 kg	6 kg[a]
Type of tumble dryer	Private air condenser	Semi-professional air condenser	Semi-professional air condenser
Average no. cycles during lifetime for tumble dryer	900	11 000	11 000
Floor area used [m^2] per washing pillar	1	5	5
Average resource consumption for one standard load, washed and dried			
Energy [MJ]	11.18	11.92	20.79[a]
Water [L]	44.49	47.66	97.08
Detergent [g]	66[b]	85.8	132

[a]Since the size of the tumble dryer is smaller than the washing machine, it is assumed that the dryer is used 1.67 times per standard load within shared laundry 2

[b]This value differs from the EuP reports since the original value was based on IEC/EN 60456, which is not suitable for estimating environmental impacts (see Annex R.3 in IEC/EN 60456). For comparative reasons the value used in this study was instead estimated in the same way as the original values for the shared systems

(Graulich et al. 2011). Additionally, the reports identify two different washing machines that are suitable for, and common in, shared laundries (e.g., in multi-family buildings). One of these machines is described as “semi-professional” and the other one as “professional”. Only one tumble dryer is noted as being appropriate for shared use. Note that all the semi-professional and professional machines are much more robust than their private counterparts. This can both be seen in the number of wash/dry cycles the machines are able to perform during a lifetime, see Table 1, as well as the material composition of the machines themselves, see Table 2.

All three systems assume that the consumers use a tumble dryer to dry their clothes, even though only 50% of the Swedish households asked by Schmitz and Stamminger (2014) stated that their home was equipped with a tumble dryer. The reasoning behind this is twofold. First, it is unclear if the data is representative for older multi-family houses since the question might fail to capture respondents that have *access* to a tumble dryer (i.e., in a shared laundry), but not own one personally. Secondly, newly built apartments in Sweden usually are pre-equipped with a private washing pillar consisting of a washing machine and a tumble dryer. Thus, to make the result comparable between the private and shared systems, it was assumed that the residents, regardless of system, would choose to use a tumble dryer instead of drying the clothes using a drying cabinet or by hanging it on a line (either inside the building or outside).

Impacts of building usage within each system were modelled on the basis of floor area used. In the private system, it was assumed that the machines were placed on top of each other (i.e., a “washing pillar”) as is common in many apartment buildings. The floorspace used in the shared systems was estimated using floorplans for newly built shared laundry rooms in Sweden (Rotocon 2019). Here, the average

Table 2 Aggregated LCI-data for the washing machines and tumble dryers

Materials [g]	Washing machines			Tumble dryer	
EuP category	Private	Semi-professional	Professional	Private	Semi-professional
Bulk plastics	11 536	12 926	8 150	12 800	13 300
Technical plastics	298	544	1 050	652	400
Ferrous metals	33 850	32 362	176 820	23 473	40 500
Non-ferrous metals	3 804	5 398	18 030	1 364	3 500
Electronics	173	165	4 300	1 988	1 900
Miscellaneous	22 653	22 468	2 100	1 856	600
Total amount	72 313	73 863	210 450	42 132	60 200

laundry room occupied approximately 15 m^2 and contained up to three washing machines and three tumble dryers. This means that each washing pillar would need 5 m^2 floor area. The resource consumption per wash and dry cycle (energy, water, and chemicals) was taken directly from the EuP-reports (Graulich et al. 2011; Presutto et al. 2007b; PricewaterhouseCoopers 2009).

According to the reports, the average wash temperature for European consumers is 45.8 °C (Presutto et al. 2007b) but there are some indications that a shift is occurring, and younger people seem to wash at colder temperatures (Laitala et al. 2012). To account for this change, the benchmark that was used in each of the systems was the standard wash program at 40 °C. On a similar note, European households seem to load washing machines at 60–68% of the rated capacity (Presutto et al. 2007a), even though the majority of consumers state that they fill the machine to its maximum level (see for example, Alborzi et al. (2017) and Kruschwitz et al. (2014)). To not underestimate emissions, the model conservatively assumes a loading rate at 60% of the size of the washing machine within that specific system. A summary of the machine characteristics within each laundry system can be found in Table 1.

2.2.2 LCI data

The LCI data from the EuP-reports for the machines and cleaning products is illustrated in Table 2 and Table 3. Since the reports for washing machines and dryers exclude building usage, emissions associated with the building itself were taken from a cradle-to-grave LCA-report published by the Swedish Environmental Research Institute (IVL). The report is deemed representative for newly built energy efficient multi-family concrete houses in Sweden and follows CEN EN 15978 and CEN EN 15804, including product stage A1-3, construction stage A4-5, use stage B2 and B4-6, and end of life stage C1-4 (Liljenström et al. 2015). The data from the report used in this study regarding emissions from the building itself is illustrated in Table 4 for three different cases: high, moderate, or low emissions during the total lifecycle of the house. To account for these variations in data, the expected changes of GHG emissions were calculated in the sensitivity analysis for two of the building properties: (1) bigger/smaller floorspace used within the shared systems and (2) higher/lower estimated emissions per square meter floorspace. Further variations in data that have been accounted for are described in Sect. 2.3.

Table 3 Relative composition of the detergent and softener used in the model

Materials used in the detergent	[%]	Materials used in the softener	[%]
Fatty alcohol sulfonate	16	Fatty alcohol sulfonate	20
Soap	7	Fatty acids, from vegetarian oil	4
Zeolite	22	Sodium perborate, monohydrate	2
Sodium silicate	4	Water, demineralized	74
Sodium percarbonate	7		
Sodium sulfate	24		
Sodium perborate, tetrahydrate	20		

2.3 Sensitivity analysis

In addition to expanding the analytical framework for domestic laundry, this study also aimed to investigate some of the dynamics within the system itself. To do this, a sensitivity analysis for the following factors was calculated:

Temperature—In order to quantify the change in GHG emissions from a changed wash temperature, the model was run using the energy consumption associated with washing at 30 °C, 40 °C, 60 °C, and 90 °C.

Type of building—To understand how GHG emissions change with the type of building, emissions was calculated when situating the laundries in a building that had either low (800 kg CO_2 eq./m^2), moderate (1 000 kg CO_2 eq./m^2), or high (1 400 kg CO_2 eq./m^2) emissions per square meter (Liljenström et al. 2015).

Room size—Variations in GHG emissions as a function of room size was calculated by using a large (25 m^2 heated floorspace), medium (15 m^2 heated floorspace), or small (10 m^2 heated floorspace) laundry room for the shared laundries.

Energy versus detergent—Previous authors have demonstrated a great variability in GHG emission from domestic laundry. Expected emissions varied by as much as a factor 6.5 between European countries, and by a factor 3.5–5 within each country (Shahmohammadi et al. 2017). The variability was mainly dependent on whether the electricity supply in the country in question was based on non-renewable resources, and if the amount of detergent used per wash was high (as often is the case if the region has very hard water).

However, according to the Sinners Circle concept (Sinner 1960), the washing result is a combination of mechanical stress, chemical use, temperature, and time. In addition, the

Table 4 Emissions associated with the building for three types of energy efficient houses over a lifespan of 50 years. The values represent the sum of emissions from cradle to grave for the whole building (Liljenström et al. 2015)

Emissions from floor area used	High emissions	Moderate emissions	Low emissions
kg CO_2-eq./m^2	1 400	1 000	500

effectiveness of hot water and of some of the components in today's detergent formulas varies with temperature. Allowing for higher temperature when washing often means better cleaning results for many stains, which in practice means that a smaller amount of detergent could be used for warmer wash cycles. Tests performed by Stamminger et al. (2005) indicate that the same cleanliness can be achieved in a 90 °C program with 50% of the rated detergent dose, in a 60 °C program with the rated detergent dose, or in a 40 °C program with a 150% of the rated detergent dose. Another recent test of detergent efficiency showed that for lightly soiled clothes, the same cleanliness could be expected for a 40 °C program, even if the amount of detergent used was reduced to that of 2/3 of the recommended amount (Laitala and Eilertsen 2009). These four different scenarios were therefore used to investigate the emissions associated with the tradeoff between energy and detergent.

2.4 Sweden in the European context

To investigate the energy versus detergent tradeoff, the outcomes of GHG emissions were calculated for the private laundry when using two different countries representing European extremes for electricity supply (Poland and Sweden), as well as the European average. The amount of detergent was in turn assumed to follow the recommendations for soft water or hard water, as to better encompass the limits of variability. The resulting changes in GHG emissions can thus be used to outline the extremes, as well as the average, trade-off between energy usage and detergent dosage.

2.5 Other modeling choices

This study is focused on Swedish consumers and relies on Swedish and European data. This makes it natural to choose the Environmental Footprint EF 3.0 as the method for impact assessment (European Commission 2013), since it is developed by the European Commission's Joint Research Centre (JRC IES) for environmental footprint of products and organizations within the union. This differs to some extent from the impact assessment done in EuP, which is instead based on characterization factors from Ecoindicator-95. Datasets used in this study include Ecoinvent 3.5 and GaBi Professional 2021, whereas the EuP uses Ecoinvent 1.2 and additional datasets found in SimaPro. The bill of materials for the products in this study is the same as in the EuP-reports.

Previous studies on domestic laundry have primarily focused on energy consumption and water use with GHG emissions being the dominant environmental impact category (see for example Morgan et al. 2018; Pakula and Stamminger 2010; Retamal and Schandl 2018; Schmitz et al. 2016; Schmitz and Stamminger 2014; Shahmohammadi et al. 2017; Yamaguchi et al. 2011). To be consistent with these precedents, and the importance of these three indicators in a laundry context, we included them in the present study. However, since laundry practices are a major contributors of phosphorus to the aquatic environment (e.g., approximately 14% of the P-load from households in the UK comes from laundry detergents (Comber et al. 2013)), this study also includes eutrophication and acidification of fresh water environments in the results.

3 Results

3.1 The role of capital goods in laundry emissions

The GHG emissions associated with domestic laundry for the three different laundry systems are shown in Fig. 2. The impact for each system is different, with the shared systems resulting in levels of emissions approximately 26% lower than the private one (190 g CO_2 eq./kg for the private laundry, compared with 147 g CO_2 eq./kg and 136 g CO_2 eq./kg for the shared laundry 1 and 2, respectively). This is mainly due to the changed impact from capital goods. To aid interpretation, the relative contributions to the emissions concerning GHG emissions are shown in Table 5. The percentage in the table is normalized against the sum of initial negative impacts, meaning that any "positive" impacts from, e.g., end of life is excluded in the normalization.

The greatest contribution to GHG emissions for the private system was emissions from capital goods (approx. 38% of total emissions), whereas for the shared systems, the greatest contributor was the use of detergent (approx. 33% of total emissions). As Fig. 2 indicates, the higher number of uses indicated in Table 1 completely offsets the additional materials indicated in Table 2. As for the rest of the impact categories, the two largest contributors to emissions were detergent (first or second largest in 75% of all impact categories) and capital goods (first or second largest in 50% of all impact categories).

The total impacts for acidification, eutrophication, water scarcity, and energy for 1 kg clothes washed and dried for each of the laundry systems are shown in Table 6.

The relative contribution from each process for the private laundry is represented in Table 7. The percentage in the table is normalized against the sum of the initial negative impact of the category, meaning that any "positive" impacts from, e.g., end of life, are excluded in the normalization.

To better understand how the building characteristics (i.e., in terms of energy efficiency, see Table 4) or the size of the laundry room impact the resulting emissions, the uncertainties for these variables (explained under 2.3) were modeled. The resulting changes in GHG emissions are

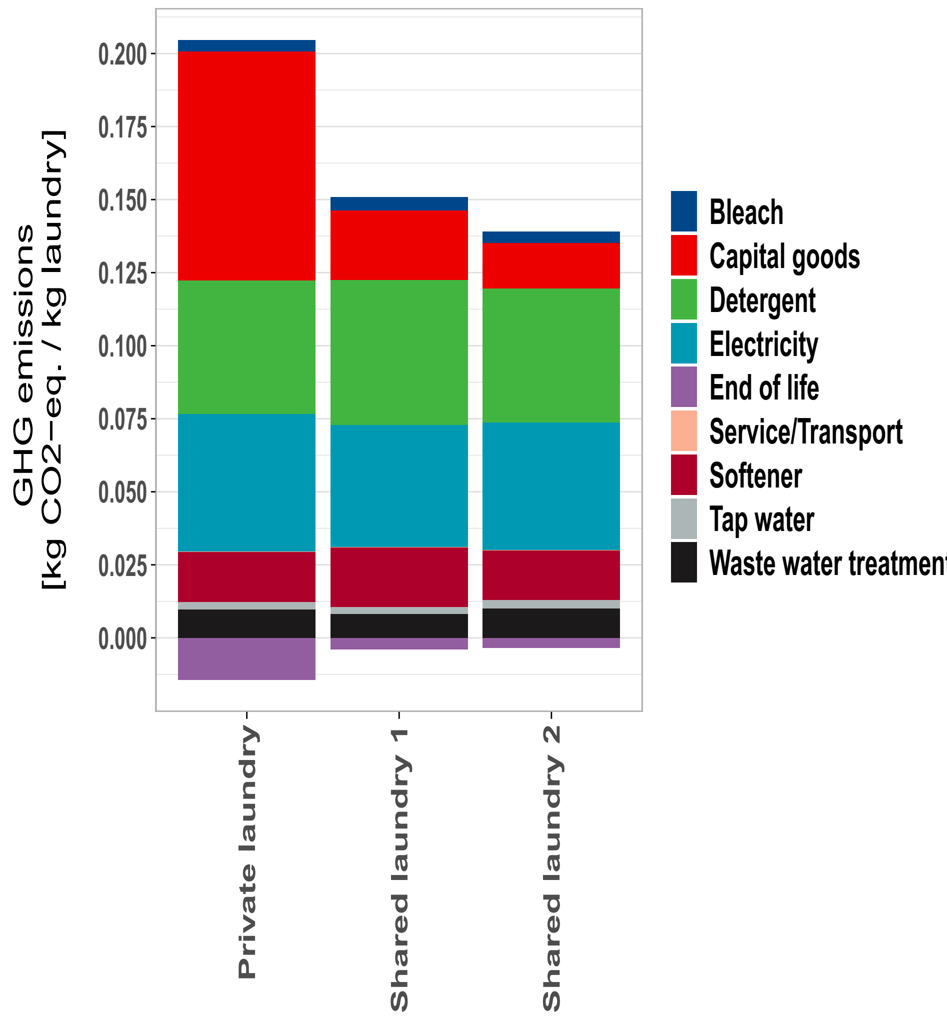

Fig. 2 Resulting GHG emissions from domestic laundry within each system

illustrated in Fig. 3. For comparison, the resulting changes are depicted together with modeled changes of GHG emissions from washing the laundry at different temperatures. It is evident that changes in the building characteristics influence the GHG emissions to a similar extent as changes in choice of wash temperature. The same goes for the size of the laundry room used. Note that the size of the laundry room only affects the shared laundry systems.

Additionally, by comparing the contribution to GHG emissions from the different capital goods, we can conclude that the building characteristics are as important as the washing machine and tumble dryer themselves, see Fig. 4. For the private laundry system, the calculated greenhouse emissions are almost equally distributed between the three sources. For the shared systems, the estimated emissions of GHG emissions from the building

Table 5 Relative contribution to GHG emissions for each of the LCA processes for one standard load, washed and dried

LCA process	Private laundry	Shared laundry 1	Shared laundry 2
Bleach	1.9%	3.0%	2.7%
Capital goods	38.3%	15.8%	11.3%
Detergent	22.3%	32.8%	32.9%
Electricity	23.0%	27.7%	31.5%
End of life	− 7.0%	− 2.6%	− 2.4%
Service/transport	0.1%	0.0%	0.0%
Softener	8.3%	13.6%	12.3%
Tap water	1.3%	1.5%	2.1%
Wastewater treatment	4.7%	5.5%	7.3%

Table 6 Total environmental impact for 1 kg clothes, washed and dried

Impact category	Acidification, terrestrial and freshwater	Eutrophication, freshwater	Water scarcity	Resource use, energy carriers
Laundry system/unit of impact category	[Mole of H + eq.]	[kg P eq.]	[m^3 world equiv.]	[MJ]
Private laundry	7.43E-04	8.58E-05	0.150	6.28
Shared laundry 1	6.72E-04	4.96E-05	0.153	5.50
Shared laundry 2	6.40E-04	5.02E-05	0.140	5.55

are approximately the same as the sum of the GHG emissions from just the whitegoods.

3.2 The trade-off between energy and detergent

The cleaning effectiveness of hot water and of many detergent components varies with temperature. Higher temperature often means better cleaning which in turn means that lower amount of detergent is needed for the same result in cleanliness. However, higher wash temperatures mean that a greater amount of energy is needed for the wash program resulting in a trade-off between the two variables in terms of environmental impact. To better understand this dynamic, the resulting changes to GHG emissions (relative to a standard 40 °C program) associated with the detergent-electricity trade-off described under Sect. 2.3 are depicted in Fig. 5 for the private laundry. As reference points, the resulting change on GHG emissions by choosing a 30 °C program or reducing the detergent dose to 2/3 of the recommended dose is also depicted in Fig. 5. Each color in the figure represents a different energy source (European averages, Sweden, or Poland). In turn, the shape of the dots represents a different amount of detergent depending on whether the water is soft (triangle) or hard (circle). All changes are normalized towards the emissions associated with a standard load for that specific energy source (i.e., the case “Normal wash, 40 °C”).

By washing at higher temperatures and with less detergent, the emissions from domestic laundry could be decreased by 6–12% in regions with hard water in Sweden (provided that the clothes do not get damaged). In areas with soft water, the net change in GHG emissions would instead be zero. For European countries using mainly non-renewable energy sources, the tradeoff would lead to increased GHG emissions. Lastly, it is interesting to note that reducing the program temperature from 40 °C to 30 °C only reduces the impact by 2-8% irrespective of the background energy source.

4 Discussion

4.1 Previous studies

The dosing of the detergent turned out to be the first or second most important factor in this study depending on the system type (private or shared). This is consistent with more recent LCAs on domestic laundry (see for example Yamaguchi et al. (2011) or Shahmohammadi et al. (2017)) but contrary to findings of previous studies for shared laundry services which concluded that the energy usage was the most important factor (Garcilaso et al. 2007). Unfortunately, since the scope of this study differs from previous LCA models for private versus shared systems for domestic laundry, it is

Table 7 Relative contribution to acidification, eutrophication, water scarcity, and resource use for the private laundry (1 kg clothes, washed and dried)

Private laundry	Acidification, terrestrial and freshwater	Eutrophication, freshwater	Water scarcity	Resource use, energy carriers
LCA process/unit of impact category	[Mole of H + eq.]	[kg P eq.]	[m^3 world equiv.]	[MJ]
Bleach	2.8%	3.1%	0.6%	1.0%
Capital goods	29.6%	49.9%	2.4%	12.0%
Detergent	32.0%	25.1%	5.7%	11.8%
Electricity	16.8%	1.0%	4.6%	67.0%
End of life	− 16.0%	0.0%	− 1.3%	− 4.2%
Service/transport	0.1%	0.0%	0.0%	0.0%
Softener	14.3%	7.0%	6.3%	6.3%
Tap water	2.3%	3.2%	80.3%	1.0%
Wastewater treatment	2.1%	10.7%	− 79.8%	1.0%

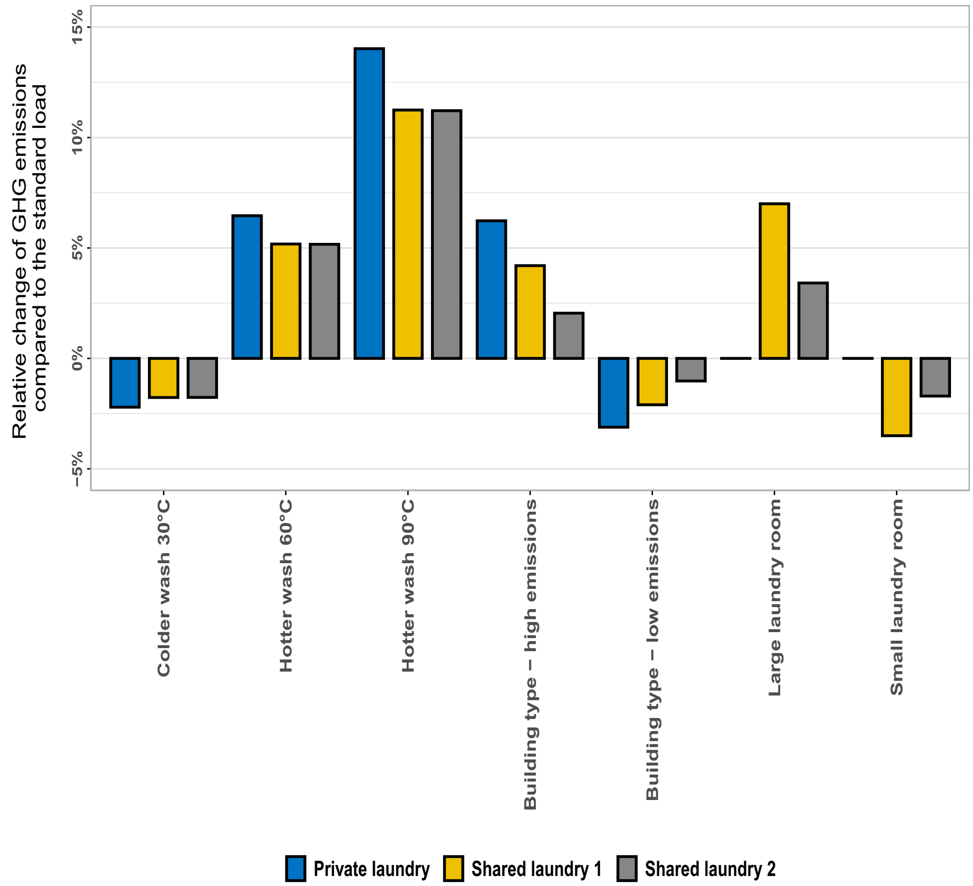

Fig. 3 Changes in GHG emissions due to different wash temperature, type of building, or size of the laundry room

hard to compare results for GHG emissions. For example, both Garcilaso et al. (2007) and Haapala et al. (2008) used Eco-Indicator Points to illustrate environmental impacts, and the study done by Amasawa et al. (2018) has a different functional unit. Re-creating the studies for comparison is theoretically possible but lies outside the scope of this study. However, the results from such a procedure would probably yield much lower impacts on GHG emissions since neither of the studies include the building use nor the drying process.

A study of laundry systems in Melbourne presented results in a comparable form to ours. Koerner et al. (2010) estimated the GHG emissions of domestic laundry to be 0.21 kg CO_2 eq./kg when drying the clothes on a drying line or 1.3 kg CO_2 eq./kg when using a tumble dryer. These results are much higher than the results in our study (0.190 kg CO_2 eq./kg) especially since our study also includes emissions from building construction and usage. This could be a result of regional variations, much like the calculations done for European countries by Shahmohammadi et al. (2017). For example, electricity production in Victoria, Australia, is heavily dependent on black and brown coal. In addition, the washing machine used in the study is somewhat larger (7.03 kg) than the machine used in our study (5 kg), while at the same time using more energy per wash cycle (5.72 MJ versus 3.44 MJ). This is also the case for the tumble dryer, where the energy consumption per dry cycle in our study was 7.74 MJ compared with 17.42 MJ in Koerner et al. (2010). Lastly, Koerner et al. (2010) write that "*Australian detergents are specially formulated for local conditions and are significantly different compared to other parts of the world*". However how, and to what extent, this has influenced the result is unclear without access to the model data.

4.2 Variability and uncertainty

4.2.1 Data sources

A common challenge for many LCA studies is to account for uncertainties and variability in data as well as in the models used (Ascough et al. 2008). The basic LCA model used in this study is mainly based on Swedish and European average values. Since these values are taken from, and used in similar way as, the European Commission's own preparatory studies of Eco-design requirements of EuPs, the validity of the data is assumed to be good. On the other hand, the EuPs for washing machines and dryers were finalized during 2005–2011, and since technological advancements

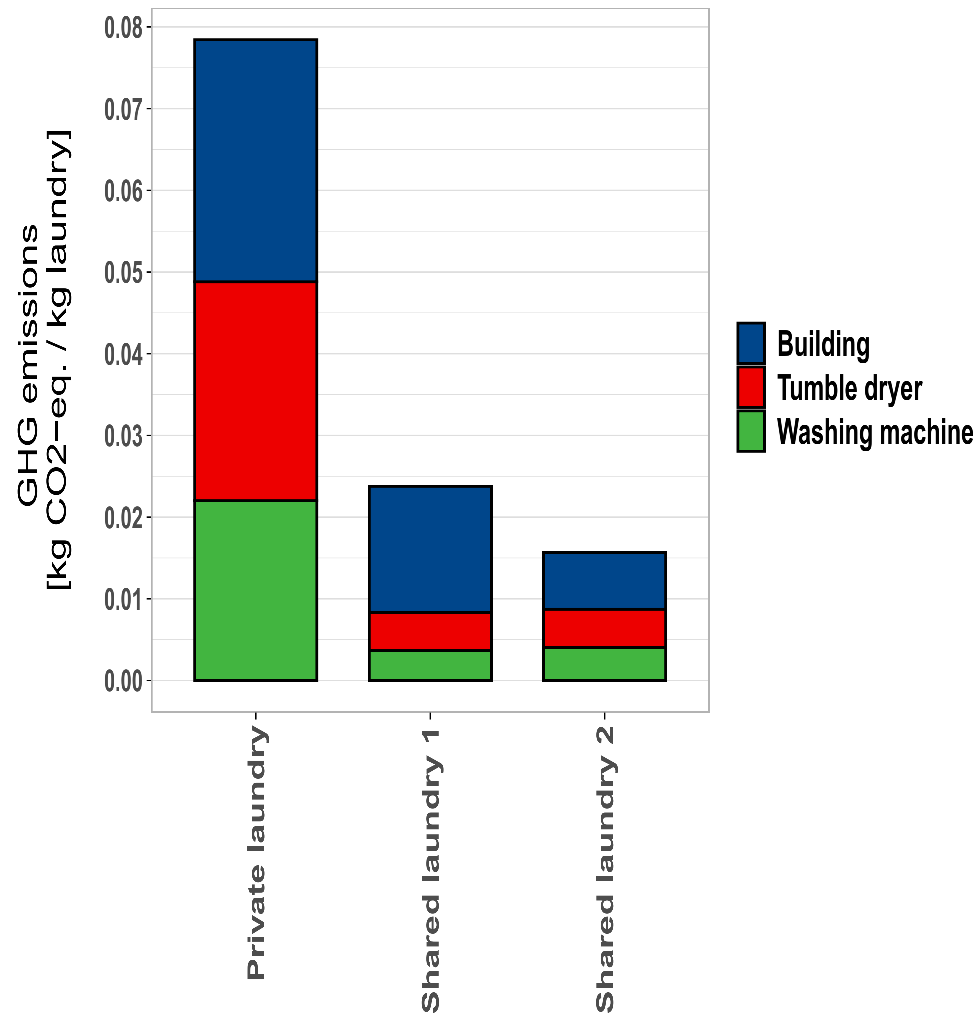

Fig. 4 Calculated GHG emissions from capital goods

are proceeding within the washing sector, emission estimates should be viewed as indicative of previously installed whitegoods rather than the latest equipment. This relates not only to changes in resource consumption efficiency for running newer machines, but also to changes in size for domestic washing machines. For example, a previous study by Schmitz et al. (2016) has shown that the average rated capacity of newer private washing machines sold within EU is growing, from 5 kg in 2003 to 7.5 kg in 2014. How these changes in size might affect resulting emissions would be interesting to further investigate but are outside the scope of this study. In any case, being aware of these changes indicate that the results from this simulation are more suitable for comparative uses between systems, rather than seen as absolute values for current emissions from domestic laundry in Sweden.

Another uncertainty concerns the emissions associated with the background system of the building. Although the values used in our model (500–1400 $kgCO_2e/m^2$) are lower than previous studies, the differences are small. For example, a recent international review of 95 residential buildings found that embodied carbon emissions varied between 179 and 1050 kg CO_2e/m^2 and for the operating phase between 156 and 4050 kg CO_2e/m^2 (Chastas et al. 2018). In France, the total emissions for multi-family buildings have been estimated to somewhere between 575 and 1910 kg CO_2e/m^2 (Hoxha et al. 2017). Swedish examples include values for a wooden single-family house with a total emission of 567 kg CO_2e/m^2 for stage A1-A5, B1-B7, and C1-C4 (Petrovic et al. 2019). Lastly, Andersson et al. (2018) found that embodied emissions for a newly built multi-family house in Sweden are approximately 391 kg CO_2e/m^2 for module A1-A5.

Even though the data is newer, the emissions associated with square meter floor area used during the whole life cycle of the house only accounts for GHG emissions (Liljenström et al. 2015). This means that all the other indicators are underestimated in the model. Additionally, the emissions calculated in the report are based on a recent, low-energy apartment building in Sweden, with an expected lifetime of 50 years. This introduces uncertainty

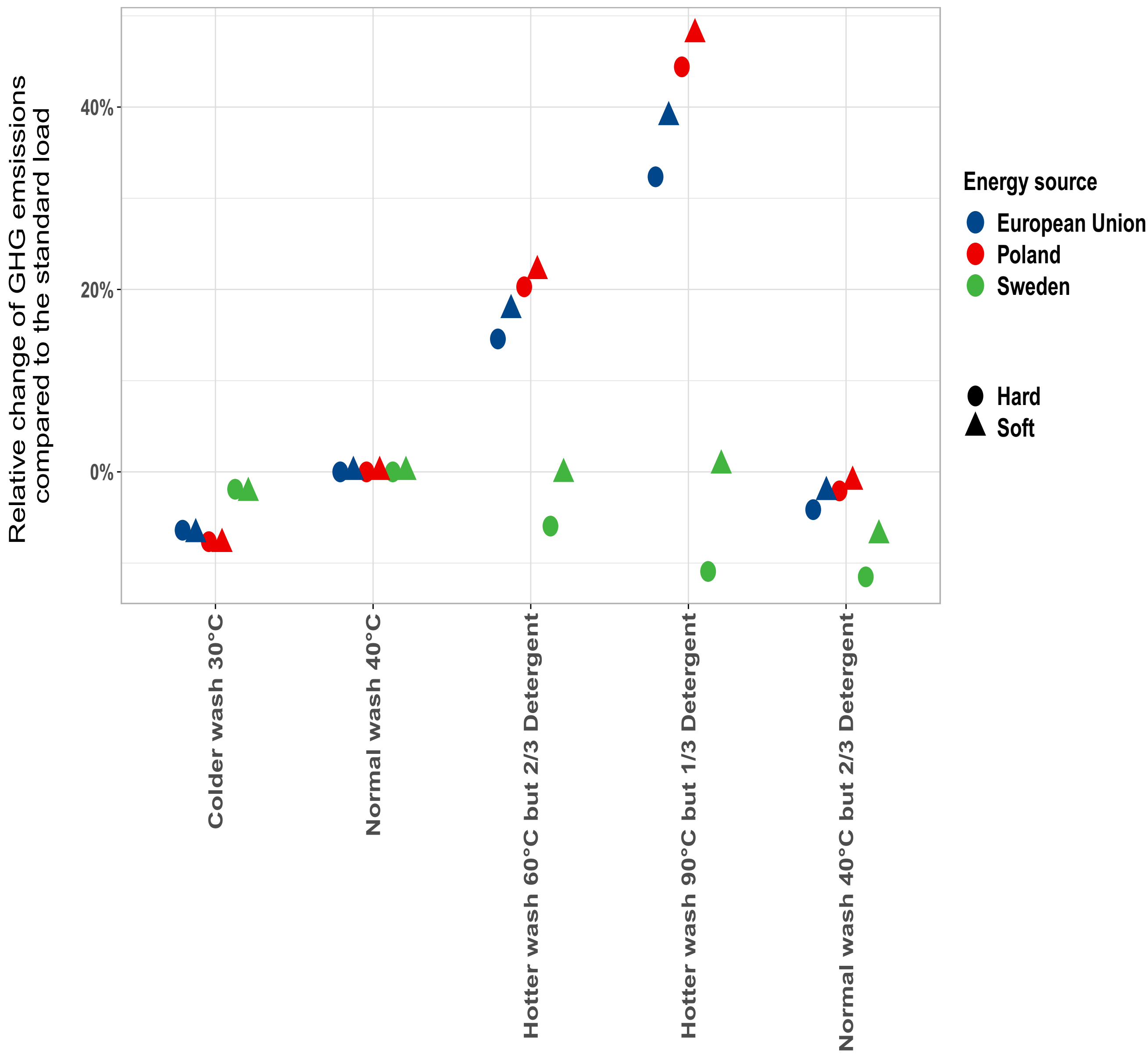

Fig. 5 Resulting change of GHG emissions for trade-offs between detergent and temperature. Uncertainties affecting the result are mainly detergent dose as a function of water hardness

to the model when comparing with European and Polish data since the building may not be representative of current average European or Polish buildings. However, since we used Swedish data for heating and energy use, the GHG emissions from the use phase are likely to be a conservative estimate and lower than European and Polish averages (due to the dominance of renewable electricity sources in Sweden). Consequently, despite its large contribution to overall impacts, the relative contribution to GHG emissions from building use should be seen as a best-case scenario in the context of current European and Polish housing stock. However, due to increased EU regulations on building efficiency, it is not unreasonable to view the contribution to GHG emissions from the building as a proxy for newly built houses in Europe. Despite this low starting point for building use, the large contribution to overall GHG emissions from capital goods raises the question whether previous authors have underestimated the emissions associated with domestic laundry. This may have occurred because the manufacturing of appliances and building space is typically considered to be relatively small in comparison with the user-phase in the LCA, and thus often excluded from the analysis (Shahmohammadi et al. 2017).

It should also be noted that the datasets used for the calculating GHG emissions were the most up-to-date available in the Gabi Professional database in 2021. However, in practice, the Polish electricity mix used represents data from 2015 and should therefore be considered a "cornerstone scenario" (Pesonen et al. 2000). On the other hand, according to the IEA, the carbon intensity of Polish electricity changed little between 2015 and 2018, having improved considerably since 2006 (IEA 2021). As countries such as Poland wean their electricity sectors off coal and the transition to renewables continues, the relative significance of electricity supplies shown in our results can be expected to reduce in comparison with processes with considerable mineral carbonate or crude oil origins (e.g., building materials and detergents, respectively).

4.2.2 Detergent recipe

It is important to note that the detergent used in the study does not account for the great variety of detergent recipes on the market, but rather represent a generic recipe for the composition based on the EuP-reports. An assessment of the influence of detergent formulation is not within the scope of this LCA but may present an interesting horizon for future research. Likewise, the LCA model does also not include other types of detergent-free machines, e.g., washing with de-ionized water. Since the detergent is such a dominant contributor to all the impact categories, more research is needed to see how these types of alternative systems could influence the environmental impacts from domestic laundry.

4.2.3 Consumer behavior

Regarding consumer behavior, the data used in the model has its origins in self-reporting. This source of behavioral information is characterized by uncertainty since humans tend to have problems remembering the decisions and frequencies of past actions, especially for habitual behaviors (Verplanken et al. 2005). This means that there are some uncertainties regarding the amount of detergent used, whether the machine is fully loaded, and how often the machines are used. For example, this study assumes a linear relationship between dosage and machine size which might not reflect how people behave in real life. Especially since only a fraction of the European consumers state that they follow the recommendations provided on the detergent packaging (Alborzi et al. 2017).

Additionally, the reference flow in the model is 1 kg of laundry cleaned and dried. The calculations were performed by first calculating the emissions from one cycle of washing and drying, and then weighting the results with the amount of laundry in the machine (i.e., 60% of the rated capacity of each type of washing machine). Since this loading rate is an average value for European consumers, it might differ substantially between different individuals and accounting for a different filling percentage would lead to changed emissions per kg laundry. Should the filling percentage also differ between private and shared laundries (e.g., because of availability, time constraints, or direct costs such as for coin-operated machines), the difference between the systems would change accordingly. Higher loading rates for shared laundries would tend to strengthen our conclusion regarding the significance of the impacts of whitegoods manufacturing and building space used. With that said, it should also be noted that this model assumes that the consumer *either* washes using a private laundry or a shared facility since this historically has been the case for Sweden. This might not be true in other countries and a recent article by Moon (2020) showed that some consumers instead use both of the alternatives, leading to lower filling rates and higher emissions per kg laundry washed.

Lastly, changing from one type of system to another might influence other types of consumer behavior not necessarily included in the functional unit. This in turn might affect the overall emissions from doing laundry but are not included in the LCA model. Although speculative, changes in behaviors might be higher or lower wash frequency (e.g., washing training clothes directly after use) or changes in choice of drying (e.g., line drying inside your own apartment rather than using the communal tumble dryer).

4.3 Implications for policy

4.3.1 Shared versus private

According to estimations from housing developers, approximately 80–100% of all new multi-family buildings built in major cities in Sweden since 2012 were equipped with privately owned machines, rather than a shared laundry room (Borg and Hogberg 2014). Looking at the result in this study makes it clear that it would be beneficial from an environmental point of view to introduce policies to reverse that trend. According to Statistics Sweden, approximately 193,300 apartments were completed in multi-family buildings throughout Sweden during the year 2012–2018 (Statistics Sweden 2019). If we assume that each apartment runs 5 washing programs a week (the average wash frequency in Sweden according to Presutto et al. (2007b)), this translates roughly to an annual emissions of 2500 metric tonnes CO_2-eq. per year in 2019 (and each year thereafter) that could have easily been avoided, just from domestic laundry in Sweden.

It is also important to mention that reclaiming the shared laundry room concept for multi-family houses would not only limit the emissions from domestic laundry but could also lead to more efficient use of building space. This would mean that there could be economic incentives for shared laundries (especially in densely populated cities), provided that the laundry rooms are designed in an appealing way for consumers (Amasawa et al. 2018).

4.3.2 The role of detergent

Policies concerning domestic laundry in Sweden and in Europe have historically focused on making laundry machines more energy efficient. This has in turn resulted in huge technological advancements over the last decade (Graulich et al. 2011; Presutto et al. 2007a), marketing campaigns (Morgan et al. 2018), and behavioral shifts (Laitala et al. 2012) towards washing at lower temperatures. However, the findings in this study suggest that Sweden are starting to reach a point of diminishing returns regarding lowering emissions using energy efficiency measures, as well as for measures targeting lower wash temperature. Looking at areas with hard water, to further reduce the emissions from domestic laundry, it might actually be preferable if some consumers *increased* the average washing temperature, provided that this shift was coupled with a reduction of the amount of detergent used. Or in other terms, since the current environmental labeling systems for washing machines in Europe are mainly focused on energy consumption rather than emissions, an energy efficient washing machine/program run in Sweden could actually lead to higher levels of emissions than a "normal" machine/program run at a higher temperature (but with less detergent).

However, these recommendations are foremost a result from the low GHG emissions associated with the electrical grid mix in Sweden. For other European countries, such a change in behavior would instead lead to higher emissions from domestic laundry. With that said, since many European countries are striving for greater use of renewable energy sources, this situation will change. One possible way to better understand when such a point has been reached would be to expand the system boundary for the environmental labeling systems in Europe so that it also includes possible tradeoffs between variables that are codependent (e.g., temperature and detergent). Another possible initiative would be to inform and educate consumers within countries with mainly renewable energy sources (e.g., Sweden) about this dynamic so that they themselves can make informed decisions.

5 Conclusions

GHG emissions from private laundries in Sweden were estimated to be 190 g CO_2 eq./kg laundry (washed and dried). If a shared laundry was used instead, the resulting emissions decreased by approximately 26%, mainly due to avoided emissions from capital goods. Because of this, it is evident that capital goods play a much bigger role for environmental impacts from laundry than previously thought. This is important not only for GHG emissions: capital goods were among the two largest contributors to emissions in more than 50% of all the impact categories. The finding is especially important for doing laundry using privately owned machines, where capital goods contributed approximately 38% of the estimated GHG emissions. As a comparison, in the shared systems, capital goods contributed approximately 11–16% to the estimated amount of GHG emissions. Looking at capital goods in more detail, the use of the building was at least as important as the machines themselves.

The above results make it possible to assess the changes in expected emissions from domestic laundry for newly built multi-family homes in Sweden since the shift in the 1990s. Had these apartment buildings been equipped with shared laundry rooms instead of privately owned machines, the estimated emissions from the laundry practices of the apartment dwellers would have been reduced by 26%.

Modeling the tradeoff between wash temperature and detergent dose also yielded interesting results. First, the most important variable in the model was the amount of detergent used per cleaning cycle (washing and drying), which was consistently being the most or second most important contributor to 75% all impact categories. Additionally, since the cleaning effectiveness of hot water and of some of the components in today's detergent formulas varies with temperature, emissions from domestic laundering could for some consumers be reduced by washing at *higher* temperature but with less detergent. This pattern could be seen in Sweden within regions with hard water where the emissions from domestic laundry could be decreased by 6–12%. However, this result is foremost a consequence of the low GHG emissions associated with the electrical grid mix in Sweden. For many other European countries, such a change in behavior would currently instead lead to higher emissions from domestic laundry.

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